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Human health risks of Pb and As exposure via consumption of home garden vegetables and incidental soil and dust ingestion: A probabilistic screening tool

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ABSTRACT

The consumption of home grown vegetables may represent a significant exposure pathway for arsenic (As) and lead (Pb) relative to direct incidental ingestion of soil, thus a probabilistic screening tool for estimating these exposures was developed using regression models relating co-located soil and home garden (HG) vegetable concentrations of Pb and As established from multiple independent studies and 2-dimensional Monte Carlo analyses. For high-quantity consumers of HG vegetables (*i.e.*, the upper 95th percentile of consumers in the general population), the HG consumption pathway can be as significant as incidental soil and dust ingestion for inorganic As and, therefore, should be considered when developing generic health-based soil criteria in residential settings. Predicted Pb Hazard Quotient (HQ) estimates among young children resulting from HG consumption were 4- to 10-fold lower than exposures resulting from direct incidental soil and dust ingestion. The difference in soil/dust ingestion rates used to characterize young children (the 95th percentile of 202 mg/d) versus a lifetime residential receptor (the 95th percentile of 30 mg/d) was a primary factor contributing to the relative differences observed between HQ and incremental lifetime cancer risk (ILCR) resulting from these two exposure pathways for lead Pb and inorganic arsenic As, respectively.
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1. Introduction

The extent to which contaminated lands should undergo remediation is, in part, determined by the potential for that land to adversely impact the surrounding environment and/or human health. Effects-based soil quality criteria or standards (soil concentrations considered protective of the surrounding environment and/or human health under a given set of conditions) are typically developed in North America and Europe by regulatory agencies using standard risk assessment approaches (Health Canada, 2004a; Health Canada, 2009a; CCME, 2006; U.S. EPA, 1989; MOE, 2011) and are often used to help guide remediation and management of contaminated lands.

Soil standards protective of human health often include direct exposure pathways, including incidental soil and dust ingestion, direct dermal contact and the inhalation of soil-born particulate matter. The uptake of metals from soils by garden vegetables (and their subsequent consumption) has been recognized as a potential indirect human exposure pathway and is a function of both the transfer of elements from soil to garden produce, and the consumption rate of that produce. Although the HG produce consumption exposure pathway is considered (typically estimated through the use of plant uptake factors) within larger-scale risk assessment protocols such as those promulgated by the U.S. EPA (2005), there are generic health-based soil standards (such as those developed by the MOE (2011)) that have not included this exposure pathway due to the high degree of uncertainty that exists with regards to a number of pathway specific assumptions (for example, plant uptake factors, produce consumption rates, food preparation methods, etc.) and, therefore, the question remains: does the consumption of home grown vegetables represent a significant exposure pathway for As and Pb relative to direct incidental ingestion of soil?

Incidental soil ingestion rates, particularly among young children, have the potential to greatly influence chemical exposure estimates and hence, the overall health-based soil standard. As such, investigations of soil and dust ingestion rates among young children have gained considerable attention in the past and have led to numerous recommendations with respect to point-estimate values for soil ingestion. The Child-Specific Exposure Factors Handbook (U.S. EPA, 2008a) recommends a central tendency total soil and dust ingestion rate among young children (1 to <6 years) of 110 mg/d (50 mg/d soil plus 60 mg/d indoor dust). The U.S. EPA (2008a) has classified overall confidence in this recommendation as 'low', indicating that the data upon which the recommendations are based are highly uncertain. Health Canada (2004a, 2009a) and the CCME (2006) recommend a soil ingestion rate of 80 mg/day for young children (7 months to 4 years) and 20 mg/day for all other age groups. The Ontario Ministry of

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the Environment (MOE, 2011) applies a soil ingestion rate of 200 mg/d for young children during the derivation of generic health-based soil standards. These types of point-estimates do not quantitatively address the variability and uncertainty surrounding estimates of long-term soil ingestion rates.

The objectives of the current study were to first develop a probabilistic screening tool, enabling risk assessors to express upper percentiles of Pb and As health risk estimates (and their associated uncertainties) resulting from the consumption of HG produce, and soil itself, as a function of soil concentration. The second objective was to examine (quantitatively) the inter-variability and parameter uncertainty of human health risks associated with Pb and inorganic As exposure resulting from the consumption of home-produced vegetables and incidental soil ingestion. To achieve these objectives, regression models relating co-located soil and HG produce concentrations of Pb and As established from multiple independent studies (Bacigalupo and Hale, 2011), were used in combination with standard probabilistic human health risk assessment methods developed by various regulatory bodies (U.S. EPA, 2001, 2003; Health Canada, 2004a, 2009b; CCME, 2006), Microsoft Excel® and Crystal Ball® 7.01 to facilitate a series of 2-dimensional Monte Carlo analyses (2-D MCA).

The U.S. EPA (2001) recommends, to the extent possible, that the variability in exposures among members of a given population ('inter-variability') and the uncertainty associated with a particular input parameter within a given model ('parameter uncertainty') be kept separate when conducting a probabilistic risk assessment. For the purpose of the current exercise, uncertainty refers to the lack of knowledge with regard to a specific parameter, model or value (for example, concentration of contaminant in an environmental medium) whereas variability refers to diversity or true heterogeneity within a given population. Increasing sampling will not reduce variability (U.S. EPA, 2001).

2. Methods

2.1. Quantitative risk assessment: problem formulation (conceptual model)

Two exposure pathways were considered: the consumption of HG produce, and incidental ingestion of soil and indoor dust. The characteristics of a young child (aged 7 months to 4 years) were selected to evaluate exposure to Pb while the characteristics of a residential lifetime receptor were used to assess the lifetime average daily exposure to inorganic As. The general exposure scenario was that of a typical residential setting, whereby individuals (young children and the lifetime residential receptor) were assumed to spend all their time at home while consuming a proportion of their total daily vegetable intake from a home garden.

2.2. Quantitative risk assessment: exposure assessment

2.2.1. Receptor characterization

Young children and the lifetime residential receptor were quantitatively characterized using data from regulatory agencies (Health Canada (2004a), CCME (2006), U.S. EPA (1997, 2001, 2003, 2008a, b), and from peer-reviewed literature (Richardson, 1997; Stanek et al., 2001). Consistent with the U.S. EPA (1997), the residential lifetime receptor was assumed to have an occupancy period of 33 years, representing the 95th percentile of residential occupancy durations. The lifetime receptor encompassed five life stages, namely infant, toddler, child, teen, and adult. Each life stage was amortized according to its age duration as per Health Canada (2004a) (Table 1). The human receptor characteristics used to facilitate the exposure assessment, including both point estimate values and probability distribution functions (PDFs), are provided in Table 1. Human receptor characteristics including body weight, incidental soil ingestion (for young children only), and HG vegetable intake rates were characterized using PDFs defined by Crystal Ball® 7.01, according to the distributional shape and parameters provided in Table 2. All other human receptor input parameters were considered point-estimate values and, therefore, remained constant throughout each MCA.

2.2.2. Derivation of incidental soil and indoor dust ingestion rates

A non-continuous empirical PDF (including estimates of uncertainty) describing the long-term (annual) average soil ingestion distribution among young children who took part in a mass-balance study at Anaconda, Montana as developed by Stanek et al. (2001),

Table 1
Point estimate receptor characteristics.

Receptor characteristics	Units	Infant[1]	Toddler	Child	Teen	Adult	References
Age	yr	0–6 moths	7 mo–4 yr	5–11 yr	12–19 yr	>20 yr	Health Canada, (2004a)
Exposure duration	yr	0.5	3.5	6	7	16	Health Canada, (2004a); U.S. EPA, (1997)
Amortization (duration/lifespan)	Unitless	0.0067	0.047	0.08	0.093	0.213	Calculated[2]
Exposure frequency	days/yr	365	365	365	365	365	Health Canada, (2004a)
Averaging time	days	182.5	1277.5	2190	2555	5840	Calculated[3]
Body weight	kg	8.2	16	33.6	59.7	63.1	Richardson et al. (1997)
Incidental soil ingestion rate	g/d	0.02	0.035	0.038	0.02	0.02	Stanek et al. (2001); Health Canada, (2004a)
5th percentile soil intake	g/d		0.005				Stanek et al. (2001)
95th percentile soil intake	g/d		0.091				Stanek et al. (2001)
Indoor dust ingestion rate	g/d	0.024	0.043	0.047	0.024	0.024	Calculated[4]; EPA, (2002)
HG vegetable consumption rate	g/kg/day	0	1.21	1.21	1.21	1.21	U.S. EPA, (1997)
Fraction of individuals reported consuming	Unitless	0	0.165	0.165	0.165	0.165	U.S. EPA, (1997)

Shaded cells indicate those receptor parameters characterized using a PDF.
[1]Infants (0 to 6 months) were not considered consumers of home HG vegetables.
[2]Amortization for each life stage was based on the exposure duration of a given life stage divided by the recommended life span of 75 yr (Health Canada, 2004a).
[3]Averaging time was calculated as the exposure duration multiplied by the exposure frequency.
[4]Indoor dust ingestion rates were based on the default assumption provided by the U.S. EPA IEUBK model for Pb in children (U.S. EPA, 2002).

Table 2
Description of probability distribution functions used to describe receptor characteristics.

Receptor parameter	Receptor	PDF shape	Variability/Uncertainty	Defining PARAMETERS	References
Body weight (kg)	Infant	Lognormal	Variability	mean = 8.20 Std = 2.90	Richardson (1997)
	Toddler	Lognormal	Variability	mean = 16.00 Std = 4.60	Richardson (1997)
	Child	Lognormal	Variability	mean = 33.60 std = 9.30	Richardson (1997)
	Teen	Lognormal	Variability	mean = 56.20 std = 10.20	Richardson (1997)
	Adult	Lognormal	Variability	mean = 63.10 std = 11.90	Richardson (1997)
Soil ingestion (g/d)	Toddler	Maximum extreme	Variability	5th P = normal pdf 95th P = normal pdf	Stanek et al. (2001)
5th percentile soil ingestion (g/d)	Toddler	Normal	Uncertainty	mean = −0.005 std. = 0.0006	Stanek et al. (2001)
95th percentile soil ingestion rate (g/d)	Toddler	Normal	Uncertainty	mean = 0.091 std. = 0.017	Stanek et al. (2001)
HG vegetable consumption (g/kg/d)	All receptors	Gamma	Variability	5th P = 0.02 50th P = 0.46 95th P = 5.70	U.S. EPA, 1997

was fitted to a continuous PDF (Fig. 1). Stanek et al. (2001) reported a median soil ingestion estimate of 24 mg/d (sd = 4 mg/d) and a 95th percentile soil ingestion estimate of 91 mg/d (sd = 16.6 mg/d). The results of the Chi-Square test (as facilitated by Crystal Ball® 7.01) indicated that a continuous maximum extreme PDF was the best-fit distribution shape to the non-continuous empirical distribution of Stanek et al. (2001). The continuous maximum extreme distribution was then parameterized using the 5th and 95th percentile soil ingestion rates reported by Stanek et al. (2001). The uncertainty associated with each percentile was characterized as a normal PDF using the reported mean and standard deviation provided by Stanek et al. (2001) (Table 2). A comparison between the 90th and 95th percentiles of the original empirical PDF and the best-fit continuous maximum extreme PDF (Fig. 1) illustrates the suitability of the maximum extreme PDF for use in the remainder of the study.

The U.S. EPA Integrated Exposure Uptake Biokinetic (IEUBK) model provides central tendency "soil and dust" intake rates for several different age classes of children; it assumes a 45/55 split between soil

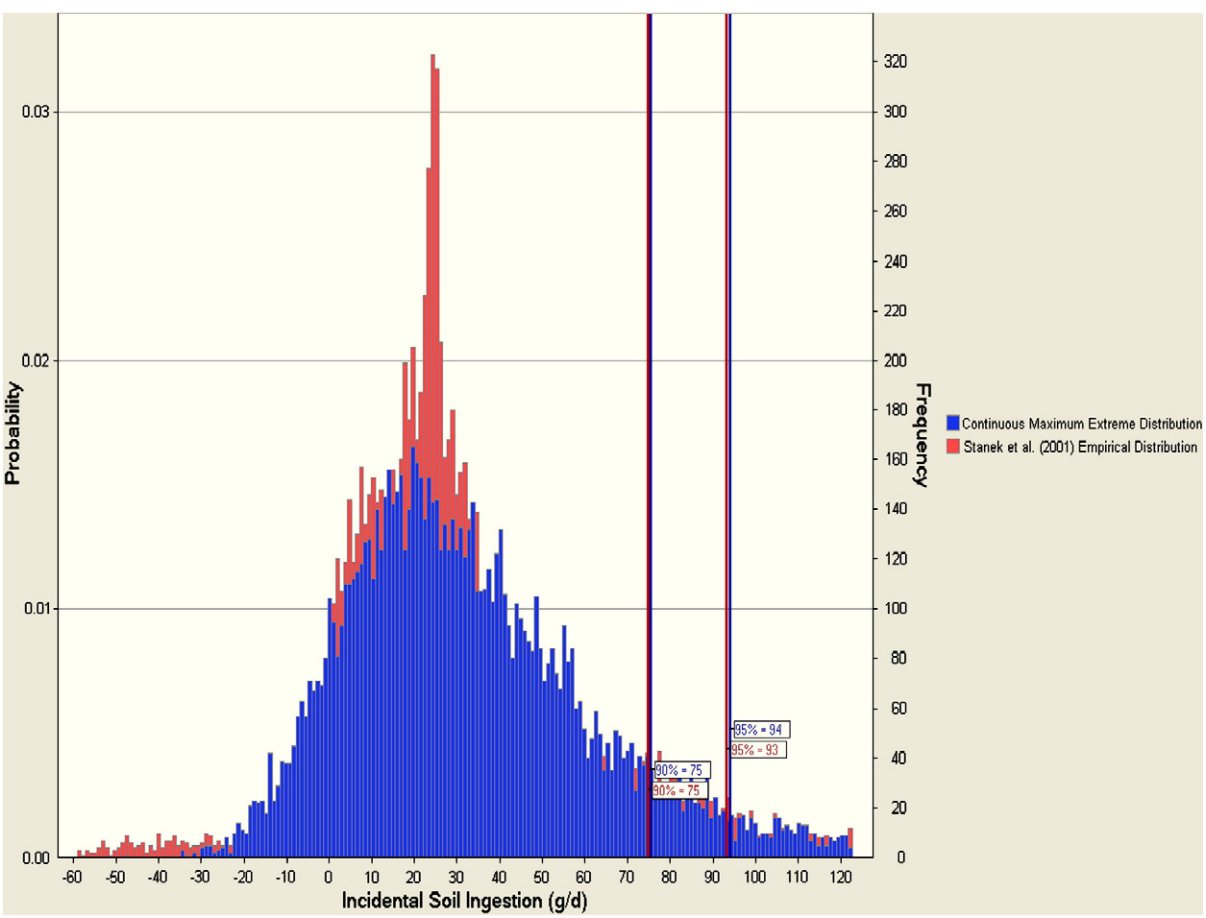

Fig. 1. An overlay diagram comparing the original Stanek et al. (2001) empirical distribution forecast and the fitted continuous maximum extreme distribution.

and indoor dust (U.S. EPA, 2002). The ingestion rates reported by Stanek et al. (2001) were for soil only, so they were assumed to represent 45% of the total "soil and dust" (soil + dust) ingestion rate. Indoor dust ingestion rates were derived as per Eq. (1), providing median and 95th percentile total "soil and dust" (soil + dust) ingestion rate of 53 mg/d (24 mg/d + 29 mg/d) and 202 mg/d (91 mg/d + 111 mg/d), respectively.

Equation 1: Indoor dust ingestion

$$indoor\ dust\ ingestion\ rate(g/d) = \left(\frac{soil\ ingestion\ rate}{0.45}\right) - soil\ ingestion\ rate \quad (1)$$

2.2.3. Derivation of HG vegetable intake rates

Seasonally-adjusted percentiles (as reported in Table 13–33 of the U.S. EPA, 1997) representing consumer-only consumption rates (g/kg body weight/day) of home produced vegetables in the northeast region of the United States (including Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island and Vermont) were used with Crystal Ball® 7.01 to fit a continuous PDF for this variable. A number of different distribution shapes including lognormal, gamma, and max extreme were applied; forecasts based on 10,000 iterations of a Monte Carlo simulation were generated for each distribution. Based on a visual comparison of upper percentiles (75th to 95th on a 5% increment) with the original seasonally-adjusted percentiles, the gamma distribution was selected as the most representative continuous PDF of seasonally-adjusted, consumer-only intake rates of HG vegetables. The 5th, 50th, and 95th percentile intake rates reported by the U.S. EPA (1997) were then used to define the gamma distribution. Individual consumer-only intakes were then adjusted to per capita intake rates by multiplying them by the fraction of individuals who reported consuming home-grown vegetables.

2.2.4. Derivation of Pb and As concentrations in HG vegetables and indoor dust

Lead and As concentrations in leafy and below-ground root vegetables were predicted using the linear regression models developed by Bacigalupo and Hale (2011). The uncertainty associated with each model was carried forward into the exposure assessment by expressing the uncertainty surrounding the mean slope (β_1) and y-intercept (β_0) estimates of each regression model as normal PDFs (Table 3). The mean slope and y-intercept values in combination with the upper and lower 95% confidence limits (*i.e.*, the 90% CI) were used to parameterize the normal PDFs. Soil concentrations were defined as single point-estimate values. As per the IEUBK model (U.S. EPA, 1998), a coefficient of 0.7 was used to describe the contribution of Pb in exterior soil to that found in indoor dust. The IEUBK default value of 0.7 is the slope of a linear regression equation describing the relationship between indoor surface dust, air and soil. The regression equation was derived using data from sites where soil was the major contributor to household dust (U.S. EPA, 1994). The coefficient describes soil's contribution to indoor dust only and does not include other potential contributing sources of Pb in house dust as would a ratio (such as those developed by Rasmussen et al., 2001) of measured indoor Pb dust concentrations versus measured Pb outdoor soil concentrations. Although the default soil-dust coefficient of 0.7 was specifically derived for Pb, the same coefficient was applied to As.

2.3. Quantifying exposures to Pb and As via ingestion of soil, dust and garden vegetables

Equations 2 through 5 in combination with receptor parameters discussed above and the chemical-specific modifying factors discussed below (Table 4) were used to develop a human health risk assessment model in Microsoft Excel® and Crystal Ball® 7.01. The general exposure equations used in the current study were based on recommendations provided by Risk Assessment Guidance for Superfund (U.S. EPA, 1989), Exposure Scenarios Document (U.S. EPA, 2003), and Health Canada (2004a).

Equation 2: Incidental soil ingestion

$$EXP_{SOIL} = \frac{SI \times C_{SOIL} \times RAF_{SOIL} \times F_{inorganic} \times ED \times EF}{BW \times AT} \quad (2)$$

Where

EXP_{SOIL}	exposure *via* incidental ingestion of soil (μg/kg/d)
SI	soil ingested per day (g/d; for toddler defined as a PDF)
C_{SOIL}	concentration of metal in soil (μg/g soil)
RAF_{SOIL}	chemical-specific relative absorption factor for ingestion of soil
$F_{inorganic}$	fraction of total As in soil which is inorganic
ED	exposure duration (yr)
EF	exposure frequency (d/yr)
BW	body weight (kg; defined as a PDF for all receptors)
AT	averaging time (d; equal to ED for the assessment of Pb)

Equation 3: Incidental indoor dust ingestion

$$EXP_{DUST} = \frac{SI \times C_{SOIL} \times 0.7 \times RAF_{DUST} \times F_{inorganic} \times ED \times EF}{BW \times AT} \quad (3)$$

Where

EXP_{DUST}	exposure *via* incidental ingestion of dust (μg/kg/d)
SI	soil ingested per day (g/d; for toddler defined as a PDF)
C_{SOIL}	concentration of metal in soil (μg/g soil)
0.7	proportion of indoor dust from outside soils (U.S. EPA, 1998)
RAF_{DUST}	chemical-specific relative absorption factor for ingestion of dust
$F_{inorganic}$	fraction of total As in soil which is inorganic
ED	exposure duration (yr)
EF	exposure frequency (d/yr)
BW	*body weight (kg; defined as a PDF for all receptors)*
AT	averaging time (d; equal to ED for the assessment of Pb)

Equation 4: Exposure *via* HG root vegetables

$$EXP_{RV} = ROOT_{CONC} \times F_{INORGANIC} \times HPVI \times F_{ROOT} \times RAF_{Food} \quad (4)$$

Table 3
Single-variable linear regression models [a].

Vegetable type	Metal	$ß_0 \pm SE$	$ß_1 \pm SE$	β_0 (90% C.I.)	β_1 (90% C.I.)
Leafy vegetable	As	−5.16 ± 0.25	0.85 ± 0.08	−5.65 to −4.67	0.70 to 1.0
	Pb	−5.15 ± 0.29	0.79 ± 0.07	−5.71 to −4.58	0.65 to 0.93
Root vegetable	As	−6.70 ± 0.42	1.05 ± 0.11	−7.57 to −5.84	0.83 to 1.28
	Pb	−3.73 ± 0.29	0.38 ± 0.08	−4.31 to −3.16	0.22 to 0.55

[a] Model: ln[plant] = $ß_0 + ß_1$ (ln[soil]); where concentrations are μg/g WW for plants and DW for soil, $ß_0$ = intercept of regression, $ß_1$ = slope of regression, and SE = Standard Error.

Table 4
Chemical-specific modifying factors.

Modifying factor description	Arsenic	Lead	References
Relative Absorption Factor—Soil (RAF_{soil})	0.5	0.6	U.S. EPA (1999); DEPA (2003)
Relative Absorption Factor—Soil (RAF_{dust})	0.5	0.6	Assumed
Relative Absorption Factor—Soil (RAF_{food})	1.0	1.0	Assumed
Inorganic proportion of total As in vegetables	0.42	NA	Schoof et al. (1999)[a]
Inorganic proportion of total As in soil	0.95	NA	ATSDR (1999)
Proportion of Pb and As in house dust relative to outdoor yard soils	0.7	0.7	U.S. EPA (1998)

[a] An average fraction of inorganic As in vegetables of 0.42 was derived based on the raw data provided by Schoof et al. (1999).

Where

EXP_{RV} exposure *via* ingestion of HG root vegetables (μg/kg/d)
$ROOT_{CONC}$ concentration of metal in root vegetables as a function of soil (μg/g fresh weight)
RAF_{FOOD} metal-specific relative absorption factor for food (1.0)
$F_{INORGANIC}$ fraction of total As in vegetables which is inorganic
$HPVI$ home produced vegetable intake rate (g/kg/d; PDF);
F_{ROOT} fraction of the total home produced vegetable intake that is root vegetable

Where

$ROOT_{CONC} = e^{(\beta_0 + \beta_1 \ln C_{soil})}\beta_0$
y-intercept of the metal-specific linear regression model for root
β_1 slope of the metal-specific linear regression model for root vegetables
C_{SOIL} concentration of metal in soil (μg/g soil)

Equation 5: Exposure *via* HG leafy vegetables

$$EXP_{LV} = LEAFY_{CONC} \times F_{INORGANIC} \times HPVI \times F_{LEAFY} \times RAF_{Food} \quad (5)$$

Where

EXP_{LV} exposure *via* ingestion of HG leafy vegetables (μg/kg/d)
$LEAFY_{CONC}$ concentration of metal in leafy vegetables as a function of soil (μg/g fresh weight)
RAF_{FOOD} metal-specific relative absorption factor for food (1.0)
$F_{INORGANIC}$ fraction of total As in vegetables which is inorganic
$HPVI$ home produced vegetable intake rate (g/kg/d; PDF);
F_{LEAFY} fraction of the total home produced vegetable intake that is leafy vegetable

Where

$LEAFY_{CONC} = e^{(\beta_0 + \beta_1 \ln C_{soil})}$

β_0 y-intercept of the metal-specific linear regression model for leafy vegetables
β_1 slope of the metal-specific linear regression model for leafy vegetables
C_{SOIL} concentration of metal in soil (μg/g soil)

Relative absorption factors (RAFs) for Pb and inorganic As in soil, dust, and food were taken from existing literature sources, to adjust for trace element bioavailability in the ingested materials, relative to that in the delivery vehicle of the toxicological study used to quantify hazard and/or risk. The US EPA IEUBK model contains default absolute bioavailability values for Pb in water and/or food vs. soil of 50% vs. 30%, respectively. Using these default values, a RAF (relative to soluble Pb in water and/or food) of 60% (30/50 = 0.6) for Pb in soil was assumed (DEPA, 2003). This RAF was adopted for use in the current study since the Pb PTDI used to characterize HQ values was based on an infant formula feeding study. Arsenic in soil is typically one-tenth to one-half as bioavailable as soluble forms of As in water; so a conservative relative bioavailability in soil of 0.5 (or 50%) was used for As (NEPI, 2000; DEPA, 2003). The As concentrations in vegetables and soil were reported as total (organic + inorganic) rather than inorganic As. The proportions of total As in vegetables and soil that would typically be in inorganic forms were estimated to ensure that the estimates of As exposure were of the same form (*i.e.*, the inorganic form) as the toxicity reference value used to characterize risk and, hence, the health-based soil quality criteria. A market basket survey of inorganic As in 40 different foods anticipated to account for almost 90% of the total dietary intake of inorganic As had an average inorganic fraction of 0.42 for vegetables (Schoof et al., 1999). Most of the As found in soils is inorganic, comprising approximately 95% of the total As in soil (ATSDR, 1999).

2.4. Quantitative risk assessment: risk characterization

Combining the continuous PDFs describing long-term exposure estimates to Pb and inorganic As with the point estimate toxicity reference values [an oral cancer slope factor of 1.7×10^{-3} (mg/kg−d)$^{-1}$ for As (Health Canada, 2004b) and a provisional tolerable daily intake (PTDI) of 3.57 μg/kg/d for Pb based on a provisional the tolerable weekly intake (PTWI) recommended by the WHO (1993)] produced PDFs describing hazard quotients (HQ) for Pb and incremental lifetime cancer risks (ILCR) for inorganic As. Eqs. (6) and (7) were used to quantify HQ and ILCR estimates for Pb and inorganic As, respectively.

Equation 6: Hazard quotient

$$HQ = \frac{EXP_{RV} + EXP_{LV}}{EXPLimit_{oral}} \quad (6)$$

Where

HQ Pb Hazard Quotient as a result of consuming HG vegetables
EXP_{RV} Pb exposure resulting from the consumption of root vegetables (μg/kg/d)
EXP_{LV} Pb exposure resulting from the consumption of leafy vegetables (μg/kg/d)
$EXPLimit_{oral}$ provisional tolerable daily intake Pb (μg/kg/d)

Equation 7: Incremental lifetime cancer risk

$$ILCR = SL_{oral} \times \left[\left(EXP_{infant} \times \frac{0.5}{75}\right) + \left(EXP_{Toddler} \times \frac{4.5}{75}\right) + \left(EXP_{Child} \times \frac{7}{75}\right) + \left(EXP_{Teen} \times \frac{8}{75}\right) + \left(EXP_{Adult} \times \frac{16}{75}\right)\right] \quad (7)$$

Where:

$ILCR$ incremental lifetime cancer risk (unitless);
SL_{oral} oral slope factor for As (mg/kg−d)$^{-1}$;
EXP_{Infant} total exposure for the infant via garden vegetable consumption (μg/kg/d);
$EXP_{Toddler}$ total exposure for the toddler via garden vegetable consumption (μg/kg/d);
EXP_{Child} total exposure for the child via garden vegetable consumption (μg/kg/d);
EXP_{Teen} total exposure for the teen via garden vegetable consumption (μg/kg/d);

EXP_{Adult} total exposure for the adult via garden vegetable consumption (μg/kg/d)

2.5. Derivation of health-based probabilistic soil quality criteria for As and Pb

Using the general iterative or "repeated runs" method recommended by the U.S. EPA (2001) for calculating preliminary remediation goals (PRGs) during probabilistic risk assessments, a series of 2-D MCA were performed for various concentrations of As and Pb in soil.

2.5.1. Arsenic

For inorganic As, an individual 2-D MCA was conducted at 2, 4, 8, 16, 32, 64, 128, or 256 μg/g As in soil, each consisting of an inner loop representing the variability in ILCR estimates and an outer loop, representing the uncertainty in either the mean linear regression models or the soil intake rates. Probability distribution functions (PDFs) assigned to characterize variability (in this case the inner loop of the 2-D MCA) were sampled 10,000 times, typically considered sufficient to resolve any numerical stability issues at the tails of each distribution (U.S. EPA, 2001). No specific guidance or recommendations with respect to the minimum number of iterations that should be used for the outer loop of a 2-D simulation were identified (U.S. EPA, 2001). The maximum number of outer loop iterations possible in Crystal Ball 7.01 is 250; however, running this number of iterations on the outer loop was not practical given the number of simulations evaluated, and the computational power, memory and time available. So, one scenario used 200 sampling iterations on the outer loop, and then the confidence limits of this test case were compared to the same scenario generated using only 25 iterations on the outer loop.

A series of 2-D trends and PDFs representing the variability and uncertainty in ILCR as a result of incidental soil/dust ingestion alone and the consumption of HG vegetables alone, were derived. Percentiles of ILCR and the uncertainties (upper and lower 95% confidence limits) associated with incidental soil/dust ingestion and HG vegetable consumption, were plotted against the total As soil concentration. Corresponding trend lines and equations were fitted to these data using Microsoft Excel®.

2.5.2. Lead

For Pb, seven separate 2-D MCAs at 30, 60, 120, 240, 480, 960, and 1600 μg/g Pb in soil were performed. A series of 2-D trends and PDFs, representing the variability and uncertainty in Pb HQ estimates as a result of incidental soil/dust ingestion alone, and the consumption of HG vegetables alone, were derived. Percentiles of HQ estimates and their uncertainties (upper and lower 95% confidence limits) associated with incidental soil/dust ingestion and HG vegetable consumption, were plotted against the concentration of total Pb in soil concentration. Corresponding trend lines and equations were fitted to these data using Microsoft Excel®.

3. Results

3.1. Lead—incidental soil/dust ingestion

Mean percentiles and uncertainty estimates (expressed as the 90% confidence interval (CI)) of Pb HQ values resulting from incidental soil/dust ingestion ranged from a 5th percentile of −0.003 (−0.01 to −0.002) at 30 μg/g Pb in soil, to a 95th percentile of 3.3 (2.4 to 4.3) at 1600 μg/g Pb in soil (Table 5). A linear function was determined to best fit a plot of Pb HQ estimates (as a result of incidental soil ingestion) versus total Pb soil concentration ($R^2 = 0.99$) (Fig. 3).

The Stanek et al. (2001) empirical probability distribution describing long-term soil ingestion rates among young children was associated with negative intake rates at the lower percentiles (1st through 7th percentile) resulting in a number of negative 5th percentile Pb HQ estimates (Table 5). Daily soil ingestion estimates are developed from studies that have obtained daily measurements of trace elements in both food input and fecal output (Stanek et al., 2001). Although it is not physically possible to have a daily soil ingestion rate less than zero, negative daily soil ingestion estimates may occur when the trace elements from food intake exceed those the in fecal output for a given day (Stanek and Calabrese, 1995). In general, negative soil ingestion estimates are not reset to zero since a bias may be introduced in the final soil ingestion estimate due to the inclusion of only the positive intake estimates (Stanek and Calabrese, 1995).

Results of a typical "25 approximations of each percentile" (*i.e.*, 25 iterations of the outer uncertainty loop of a 2-D MCA), is demonstrated for soil Pb concentration of 480 μg/g (Fig. 2). Using the data presented in Table 5, a screening tool expressing the upper percentiles of HQ estimates for Pb (and their associated uncertainties) as a function of Pb concentration in soil was developed. For example, at 400 μg/g Pb in soil, the 95th percentile Pb HQ value (as a result of incidental soil/dust ingestion) was estimated (with 90% confidence) to fall between 0.6 and 1.0 (Fig. 3).

3.2. Lead–HG vegetable consumption

The mean percentiles and uncertainty estimates (expressed as the 90% CI) of Pb HQ values resulting from the consumption of HG vegetables ranged from a 5th percentile of 7.4E−05 (4.3E−05 to 1.1E−04) at 30 μg/g Pb in soil, to a 95th percentile of 0.3 (0.1 to 0.7) at 1600 μg/g Pb in soil (Table 6). A power function was determined to best fit a plot of Pb HQ estimates (as a result of HG pathway) versus total soil Pb concentration ($R^2 = 0.99$) (Fig. 3).

No specific guidance or recommendations could be identified with respect to the minimum number of iterations that should be used for the outer loop of a 2-D MC simulation. To offer some guidance, the outcomes of 200 iterations vs. 25 iterations at a soil Pb concentration of 480 μg/g Pb were compared. The 95th, 90th, and 50th percentile estimates of HQ, using 200 iterations, fell between 0.06 and 0.3, 0.03 and 0.2, and 0.004 and 0.02, respectively. Compared to the upper percentile (95th, 90th, and 50th) HQ estimates using 25 iterations of 0.05 and 0.3,

Table 5
Probabilistic results—lead HQ estimates for young children resulting from incidental soil/dust ingestion.

Lead in soil (μg/g)	95th percentile estimates			90th percentile estimates			50th percentile estimates			5th percentile estimates[a]		
	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL
30	3.2E−02	5.9E−02	8.2E−02	2.6E−02	4.6E−02	6.5E−02	7.6E−03	1.7E−02	2.5E−02	−1.3E−02	−3.1E−03	−1.9E−03
60	8.0E−02	1.3E−01	1.7E−01	6.3E−02	1.0E−01	1.3E−01	2.2E−02	3.7E−02	5.2E−02	−2.0E−02	−7.6E−03	4.9E−03
120	1.1E−01	2.4E−01	3.2E−01	8.7E−02	1.9E−01	2.6E−01	3.1E−02	6.9E−02	9.4E−02	−3.3E−02	−1.0E−02	1.4E−02
240	3.4E−01	4.9E−01	6.3E−01	2.7E−01	3.9E−01	4.9E−01	1.1E−01	1.4E−01	1.9E−01	−7.1E−02	−2.7E−02	1.7E−02
480	6.1E−01	9.5E−01	1.1E+00	4.5E−01	7.5E−01	9.6E−01	1.2E−01	2.6E−01	3.7E−01	−1.8E−01	−6.8E−02	2.5E−02
960	1.5E+00	2.1E+00	2.5E+00	1.2E+00	1.7E+00	2.0E+00	3.9E−01	6.2E−01	8.3E−01	−2.7E−01	−6.5E−02	1.2E−01
1600	2.4E+00	3.3E+00	4.3E+00	1.9E+00	2.6E+00	3.4E+00	6.7E−01	9.2E−01	1.2E+00	−4.7E−01	−2.1E−01	4.0E−03

[a] The Stanek et al. (2001) empirical probability distribution describing long-term soil ingestion rates among young children was associated with negative intake rates at the lower percentiles (1st through 5th percentile).

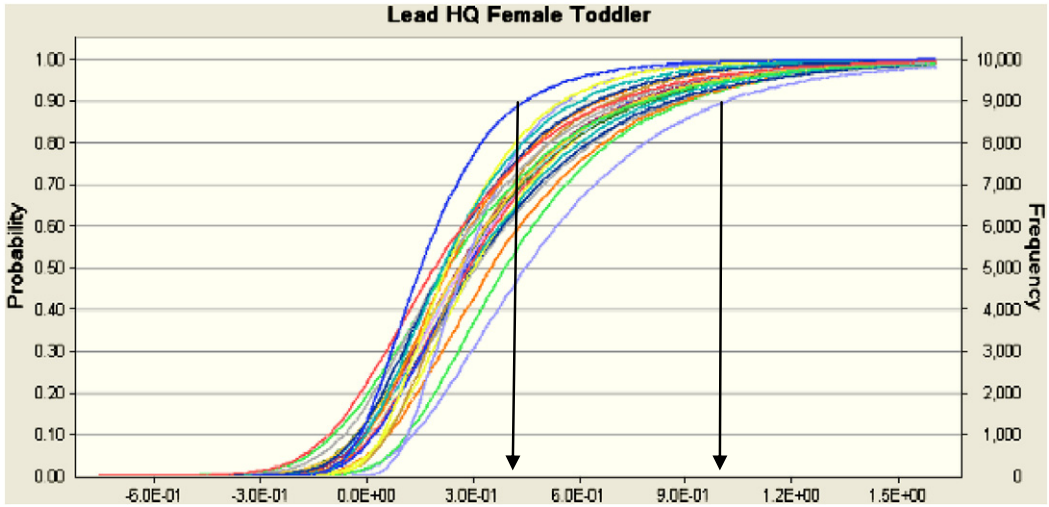

Fig. 2. Crystal Ball®—overlay diagram representing the cumulative probability distributions Of Pb hazard quotient estimates for a female toddler as a result of incidental soil/dust ingestion at a concentration of 480 μg/g Pb in yard soil—illustrating the range of 90th percentile Pb HQ estimates.

0.02 and 0.2, 0.003 and 0.02, respectively, some numerical instability may be present. Increased numerical stability may, or may not have been achievable using an increased number of iterations in the outer loop.

When these mean percentiles and associated uncertainties expressing Pb HQ values were plotted relative to soil Pb concentration (Fig. 3), it was determined that at a Pb concentration of 400 μg/g in garden soil, the 95th percentile Pb HQ as a result of consuming vegetables was estimated (with 90% confidence) to fall between 0.05 and 0.22. In other words, a Pb concentration of 400 μg/g in garden soil would not be expected to result in exposures among high-end consumers (the upper 95th percentile consumers of HG vegetables among the general population) that exceed the PTDI of 3.57 μg/kg/d due to the HG pathway alone.

3.3. Arsenic—incidental soil/dust ingestion

Mean percentiles and uncertainty estimates (expressed as the 90% CI) of As ILCR values resulting from incidental ingestion of soil/indoor dust ranged from a 5th percentile of 5.5E−07 (4.1E−07–6.5E−06) at

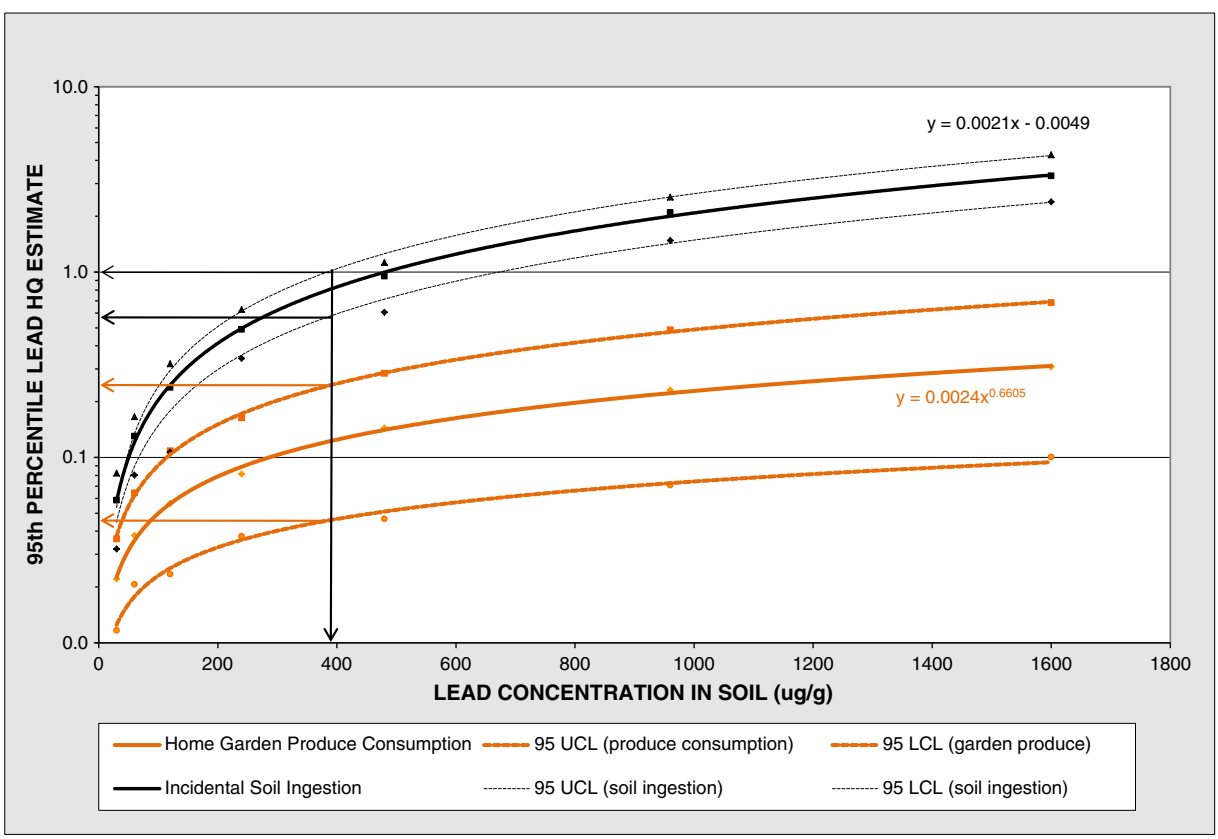

Fig. 3. A screening-level risk assessment tool proving the 95th percentile Pb hazard quotient estimates as a result of HG vegetable consumption and incidental soil/dust ingestion versus Pb concentration in soil (μg/g).

Table 6
Probabilistic results—lead HQ estimates for young children resulting from HG vegetable consumption.

Lead in soil (μg/g)	95th percentile estimates			90th percentile estimates			50th percentile estimates			5th percentile estimates		
	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL
30	1.2E−02	2.2E−02	3.6E−02	6.4E−03	1.2E−02	1.9E−02	2.0E−03	1.3E−03	7.2E−04	4.3E−05	7.4E−05	1.1E−04
60	2.1E−02	3.8E−02	6.4E−02	1.2E−02	2.0E−02	3.5E−02	1.3E−03	2.2E−03	3.6E−03	6.8E−05	1.3E−04	2.1E−04
120	2.3E−02	5.6E−02	1.1E−01	1.3E−02	3.0E−02	5.8E−02	1.4E−03	3.2E−03	6.3E−03	7.8E−05	1.9E−04	3.7E−04
240	3.8E−02	8.1E−02	1.6E−01	2.0E−02	4.3E−02	9.0E−02	2.2E−03	4.7E−03	9.2E−03	1.3E−04	2.7E−04	5.1E−04
480	4.7E−02	1.4E−01	2.8E−01	2.4E−02	7.7E−02	1.5E−01	2.5E−03	8.3E−03	1.7E−02	1.5E−04	4.9E−04	9.9E−04
960	7.1E−02	2.3E−01	4.9E−01	3.9E−02	1.2E−01	2.6E−01	4.1E−03	1.3E−02	2.8E−02	2.4E−04	7.8E−04	1.6E−03
1600	1.0E−01	3.1E−01	6.8E−01	5.4E−02	1.6E−01	3.2E−01	5.8E−03	1.8E−02	3.5E−02	3.3E−04	1.0E−03	2.0E−03

2 μg/g As in soil, to a 95th percentile of 2.0E−04 (1.7e−04–2.0E−04) at 256 μg/g As in soil (Table 7). A linear function best fit a plot of ILCR estimates (as a result of incidental soil ingestion) versus total As in soil ($R^2=1$) (Fig. 4). From this function, it was determined with 90% confidence that an increase in the As soil concentration above background of between 7.8 and 15.7 μg/g would result in an incremental increase in the lifetime cancer risk of 1 in 100,000 (as a result of incidental soil/dust ingestion) 95% of the time. The MOE (2011) estimates the 97.5th percentile of the Ontario Typical Range (OTR_{98}) of As in surface soils (not contaminated by point sources) to be 18 μg/g, resulting in a total As soil concentration of between approximately 25.8 μg/g (7.8 μg/g + 18 μg/g) and 33.7 μg/g (15.7 μg/g + 18 μg/g) that would result in a ILCR of 1 in 100,000.

3.4. Arsenic—HG vegetable consumption

As with Pb, the As-specific single-variable regression models developed by Bacigalupo and Hale (2011) were used to predict the concentrations of As in below-ground root and above-ground leafy vegetables as a function of the total As concentration in soil. Mean percentiles and uncertainty estimates (90% CI) of As ILCR values resulting from consumption of HG vegetables ranged from a 5th percentile of 8.7E−09 (5.6E−09 to1.2E−08) at 2 μg/g As in soil, to a 95th percentile of 1.7E−04 (5.9E−05 to 3.2E−04) at 256 μg/g As in soil (Table 8). The variability in HG vegetable consumption rates translated into ILCR estimates which spanned several orders of magnitude: for example, at 50 μg/g As, the 5th and 95th percentile ILCR estimates were 1.0×10^{-7} and 4.0×10^{-5}, respectively. The 95th and 50th percentile ILCR estimates and their associated uncertainties (defined as the uncertainty around the mean slope (β_1) and y-intercept (β_0) estimates of each regression model (Bacigalupo and Hale, 2011) fit a power function to the total As concentration in soil ($R^2=0.99$) (Fig. 4). From this function, it was determined, with 90% confidence, that an increase in As soil concentration (above background levels) of between 5.7 and 28.6 μg/g would result in an incremental increase in the lifetime cancer risk of 1 in 100,000 (as a result of consuming HG vegetables) 95% of the time (Fig. 4). Unlike the results for Pb, the 95th percentile ILCR estimates as a result of the incidental soil/dust ingestion and HG pathways are similar in magnitude (Fig. 4).

4. Discussion

In an attempt to answer the question: "does the consumption of home grown vegetables represent a significant pathway for As and Pb relative to direct incidental ingestion of soil?", the current study first develop a probabilistic screening tool, enabling risk assessors to express upper percentiles of Pb and As human health risk estimates (and their associated uncertainties) resulting from the consumption of HG vegetables, and soil itself, as a function of soil concentration. The second objective was to examine (quantitatively) the inter-variability and parameter uncertainty of human health risks associated with Pb and inorganic As exposure resulting from the consumption of HG vegetables and incidental soil ingestion.

4.1. Lead

The U.S. EPA screening-level soil criterion for Pb at residential sites is 400 mg/kg for bare soil in high-contact areas such as where children play. The 400 mg/kg screening level criterion was developed using the IEUBK model, along with default assumptions and exposure estimates from other background sources such as food, water, and air. The value of 400 mg/kg of Pb in soil assumes that a typical child would have an estimated risk of no more than 5% of exceeding a blood Pb (PbB) level of 10 μg/dL following direct contact with Pb-impacted soils.

In contrast, the Canadian Council of Ministers of the Environment's (CCME) Human Health Soil Quality Guideline (SQG_{HH}) for Pb of 140 mg/kg (CCME, 1999) assumes a threshold dose-response and implemented a provisional tolerable daily intake (PTDI) for Pb of 3.57 μg/kg/d based on a provisional tolerable weekly intake (PTWI) of 25 μg/kg/d, recommended by the Joint FAO/WHO Committee on Food Additives (WHO, 1993). The WHO (1993) PTWI of 25 μg/kg/d was derived using data from metabolic feeding studies of infants. The CCME Pb SQG_{HH} provides no consideration for the potential differences in Pb gut absorption between the medium used to develop the PTWI (infant feeding studies) and Pb in soil (*i.e.*, a default relative absorption factor of 1.0 was employed) and incorporates a 20% (a default "soil allocation factor") of the residual tolerable daily intake (RTDI), defined as the provisional tolerable daily intake (3.57 μg/kg/d) minus the estimated daily intake from background sources (0.39 μg/kg/d) (CCME, 1996). The

Table 7
Probabilistic results—inorganic arsenic ILCR estimates for a residential lifetime receptor resulting from incidental ingestion of soil and indoor dust.

Arsenic in soil (μg/g)	95th percentile estimates			90th percentile estimates			50th percentile estimates			5th percentile estimates		
	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL
2	1.2E−06	1.6E−06	2.0E−06	1.1E−06	1.4E−06	1.7E−06	7.5E−07	9.2E−07	1.0E−06	4.1E−07	5.5E−07	6.5E−07
4	2.8E−06	3.5E−06	4.0E−06	2.5E−06	3.0E−06	3.4E−06	1.7E−06	2.0E−06	2.2E−06	9.6E−07	1.2E−06	1.3E−06
8	8.9E−06	1.3E−05	1.6E−05	8.1E−06	1.1E−05	1.4E−05	6.1E−06	7.5E−06	8.7E−06	4.0E−06	4.7E−06	5.3E−06
16	2.0E−05	2.6E−05	3.0E−05	1.8E−05	2.3E−05	2.7E−05	1.3E−05	1.5E−05	1.7E−05	7.3E−06	9.5E−06	1.1E−05
32	3.9E−05	5.1E−05	6.2E−05	3.6E−05	4.5E−05	5.3E−05	2.4E−05	2.9E−05	3.3E−05	1.3E−05	1.8E−05	2.2E−05
128	7.9E−05	9.9E−05	1.2E−04	7.1E−05	8.7E−05	1.0E−04	4.9E−05	5.9E−05	6.6E−05	3.2E−05	3.7E−05	4.2E−05
256	1.7E−04	2.0E−04	2.2E−04	1.5E−04	1.8E−04	1.9E−04	1.0E−04	1.2E−04	1.3E−04	5.7E−05	7.4E−05	8.7E−05

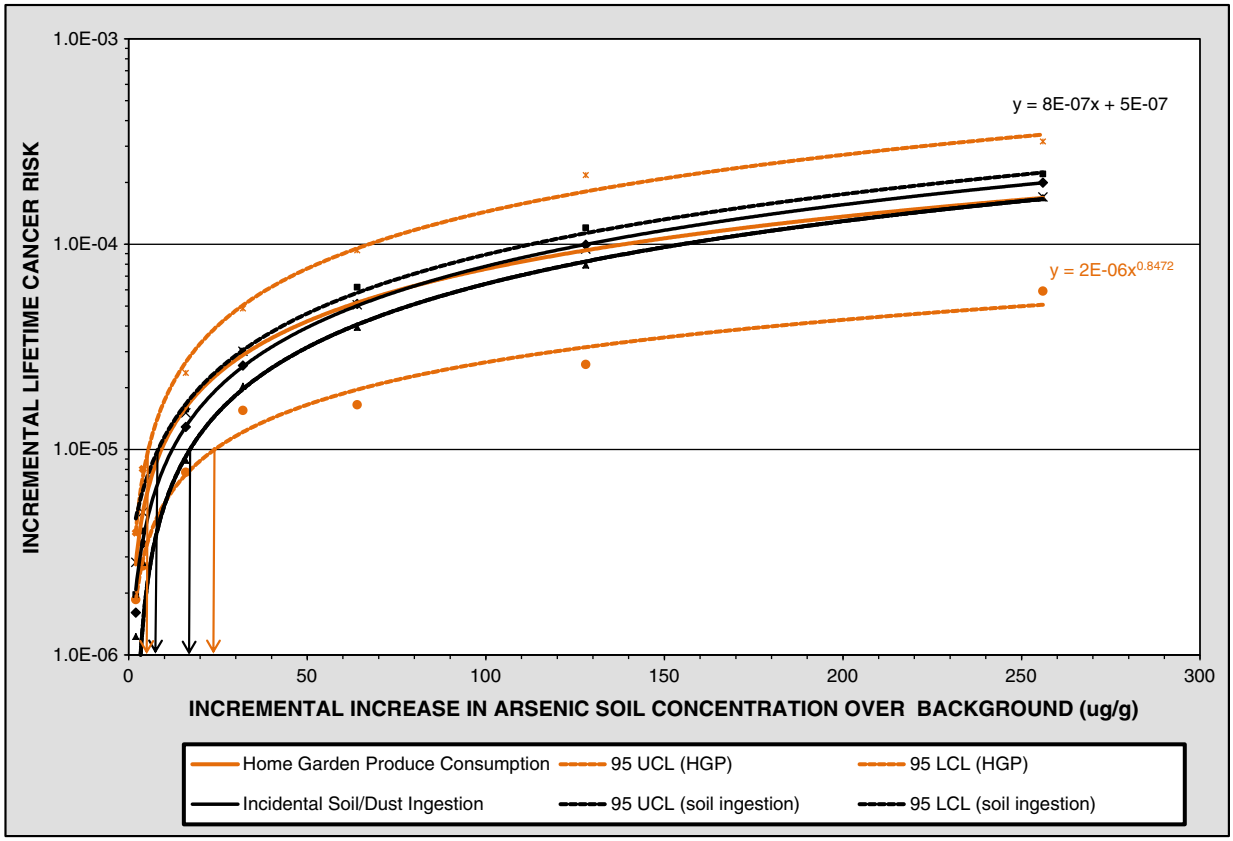

Fig. 4. A screening-level risk assessment tool proving the 95th percentile as incremental lifetime cancer risk estimates as a result of HG vegetable consumption and incidental soil/dust ingestion versus Pb concentration in soil (μg/g).

model in the present study used the provisional tolerable daily intake (PTDI) of 3.57 μg/kg/d based on the provisional tolerable weekly intake (PTWI) recommended by the Joint FAO/WHO Committee on Food Additives (WHO, 1993).

The Centers for Disease Control and Prevention's Advisory Committee on Childhood Lead Poisoning Prevention (ACCLPP) has recently made recommendations to the Centers for Disease Control and Prevention (CDC) with regards to changing the blood lead level of concern (CDC, 2012). Health Canada (2011a) has recently released a draft literature review of the toxicological and toxicokinetic data on lead in response to an increasing number of epidemiological studies showing adverse health effects below a blood lead level of 10 μg/dL. According to Health Canada (2011a,b), epidemiological studies have documented adverse health effects (including neurotoxicity, neurodegenerative, cardiovascular, renal and reproductive effects) at blood lead levels between 1 and 10 μg/dl, with no identified threshold for adverse health effects identified. The developmental neurotoxicity endpoint has been the most widely studied and according to Health Canada (2011a,b) provides the strongest weight of evidence of a causal relationship between low level lead exposure in early life and scores of intelligence (IQ) among school aged children. Health Canada (2011a,b) identified Lanphear et al. (2005) as the 'critical study' for the risk characterization of the effects of lead on children's IQ score.

This paper does not critically evaluate the strengths and weaknesses of studies used to support the conclusions drawn by Health Canada (2011a,b) and others but rather attempts to answer the question: "does the consumption of home grown vegetables represent a significant pathway for As and Pb relative to direct incidental ingestion of soil?". In this context (*i.e.*, a relative comparison between predicted health risks from two exposure pathways), the TRV applied will not change the result, so long as the TRV applied to each exposure pathway in question is the same.

Oomen and Lijzen (2004) summarized the results from a number of studies that examined the contribution of exterior soil to house dust.

Table 8
Probabilistic results—inorganic arsenic ILCR estimates for a residential lifetime receptor resulting from HG vegetable consumption.

Arsenic in soil (μg/g)	95th percentile estimates			90th percentile estimates			50th percentile estimates			5th percentile estimates		
	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL	95 LCL	Mean	95 UCL
2	1.9E−06	2.8E−06	3.9E−06	1.3E−06	2.0E−06	2.7E−06	1.5E−07	2.4E−07	3.4E−07	5.6E−09	8.7E−09	1.2E−08
4	2.7E−06	5.0E−06	8.0E−06	1.9E−06	3.4E−06	5.5E−06	2.2E−07	4.2E−07	6.8E−07	8.4E−09	1.5E−08	2.5E−08
8	7.8E−06	1.5E−05	2.4E−05	5.4E−06	1.1E−05	1.6E−05	6.8E−07	1.3E−06	2.0E−06	2.4E−08	4.7E−08	7.4E−08
16	1.5E−05	3.0E−05	4.9E−05	1.1E−05	2.1E−05	3.5E−05	1.3E−06	2.6E−06	4.1E−06	4.7E−08	9.2E−08	1.5E−07
32	1.7E−05	5.1E−05	9.3E−05	1.2E−05	3.5E−05	6.6E−05	1.4E−06	4.4E−06	8.1E−06	5.2E−08	1.6E−07	2.9E−07
128	2.6E−05	9.5E−05	2.2E−04	1.8E−05	6.6E−05	1.5E−04	2.2E−06	8.1E−06	1.9E−05	8.1E−08	2.9E−07	6.8E−07
256	5.9E−05	1.7E−04	3.2E−04	4.2E−05	1.2E−04	2.2E−04	5.0E−06	1.4E−05	2.7E−05	1.8E−07	5.2E−07	9.6E−07

The estimated contribution of exterior soil to interior house dust ranged from approximately 8 to >80%, however, the majority of studies reported that 30 to 50% of house dust originates from exterior soil (Oomen and Lijzen, 2004). At sites where Pb soil-dust coefficients were derived and where paint did not contribute significantly to dust, the range in soil-dust coefficients was from 0.09 to 0.85 (U.S. EPA, 1994). Oomen and Lijzen (2004) conclude that 30 to 70% of exterior soil present in house dust is likely a good approximation for evaluating exposure to indoor house dust.

In the absence of measured household dust data, the U.S. EPA IEUBK model recommends the application of a default soil-dust coefficient value of 0.7 (U.S. EPA, 1994). The slope of the regression line (*i.e.*, the coefficient of 0.7) represents the contribution of soil to indoor house dust and does not account for other non-soil related sources that may contribute to indoor Pb dust concentrations. Rasmussen et al. (2001) calculated geometric mean concentration ratios for several elements, including Pb [5.5 (232.6/42.3)] by dividing the geometric mean (n = 48) total metal content (*e.g.*, Pb) in house dust by the geometric mean (n = 48) total metal content (*e.g.*, Pb) in garden soil. The numerator (of the ratio) inherently includes all potential contributing sources of Pb in house dust. Rasmussen et al. (2001) concluded, in part, that the total metal concentration measured in indoor dust was often greater than that measured in outdoor garden soil and that dust generated from indoor sources may significantly contribute to exposures to certain elements, including lead, cadmium, antimony and mercury. Oomen and Lijzen (2004) recommended the application of a correction factor of two (2) to account for other non-soil related lead sources in house dust. As illustrated by Rasmussen et al. (2001) and Oomen and Lijzen (2004), other non-soil related sources of lead in house dust can be significant. However, characterizing soil's contribution to indoor dust only (as opposed to the use of a concentration ratio) was considered necessary given that the purpose of the current exposure assessment was to characterize soil-related exposures of As and Pb only so that a direct comparison between total soil ingestion (soil plus soil related dust) and garden produce consumption could be evaluated.

At a soil concentration of 400 μg/g Pb, the current screening-level model developed in this study predicted, with 90% confidence, a 95th percentile toddler HQ estimate as a result of the incidental soil/dust ingestion of between 0.55 and 1.0. Similarly, the 90% CI on the 95th percentile toddler HQ estimate from the HG consumption pathway was between 0.05 and 0.25 at a Pb soil concentration of 400 μg/g. Although the Pb HQ predictions associated with the HG consumption pathway are approximately 4 to10-fold less than HQ values resulting from incidental soil/dust ingestion, the uncertainty surrounding the HG consumption pathway (defined by the uncertainty around the best fit mean regression models) was greater than that associated with soil/dust ingestion.

The relative difference in HQ estimates between these two exposure pathways becomes greater as the concentration of Pb in soil increases. This is primarily due to the slope of ln-transformed regression equations (used to predict Pb concentrations in leafy and below-ground root vegetables) provided by Bacigalupo and Hale (2011) being less than one, suggesting that the rate of uptake of Pb by HG vegetables from soil will decrease as Pb content in soil increases. This is not an uncommon observation, as toxicity at the higher soil concentrations starts to either saturate uptake mechanisms, and/or impair plant function, including uptake of trace elements. Although the relative oral bioavailability of Pb in soil as a function of the soil Pb concentration was not investigated in the current study, it is suspected that the assumption of a constant relative bioavailability (60%) over a range of soil Pb concentrations and various soil properties, as used in the present study, might be over simplistic in nature. A number of studies, an example of which is Schroder et al. (2004), demonstrated that mammalian absorption of Pb from contaminated soils may be quite different than the absorption of Pb acetate. Recognizing this, the US EPA has issued guidance on the determination of site-specific bioavailability for Pb in soil (U.S. EPA, 2008b), and the IEUBK model now allows this input.

4.2. Arsenic

The Canadian Soil Quality Guideline (SQG) for inorganic As, under a residential/parkland scenario, is 12 μg/g (CCME, 1997). The derivation of the As SQG (CCME, 1997) assumed an acceptable ILCR of 1.0×10^{-6} (one in one million), a lifetime incidental soil ingestion rate of 20 mg/d, and an inorganic As background soil concentration of 10 mg/kg. No adjustment was made to reflect the potential differences in bioavailability between inorganic As in soil and that in water, which is the medium of the human epidemiological study upon which the U.S. EPA As slope factor was derived. Many recent studies have demonstrated that typically, the bioavailability of As in contaminated soil is not similar to that in water (Ruby et al., 1996; Rodriguez and Basta, 1999; Juhasz et al., 2007) and that the bioavailability of As might vary with particle size (Smith et al., 2009). The health-based component of the guideline (*i.e.*, the concentration of inorganic As in soil which results in an ILCR of 1.0×10^{-6}) is approximately 2 mg/kg; adding a background soil concentration of 10 mg/kg results in a SQG_{HH} of 12 mg/kg.

At 2 μg/g As in soil, the current model predicted, with 90% confidence, a 95th percentile ILCR estimate (resulting from incidental soil/dust ingestion) of between 1.2×10^{-6} and 2.0×10^{-6}. Estimates employed a generic relative bioaccessibility factor of 50% to account for potential differences in the bioavailability of inorganic As to the gut when contained in soil versus water. Similarly, the 90% CI on the 95th percentile ILCR estimate as a result of the HG consumption pathway was between 1.9×10^{-6} and 3.9×10^{-6} at 2 μg/g As in soil. At soil concentrations between 2 and 32 μg/g As, mean ILCR estimates related to the HG consumption pathway are greater than mean ILCR estimates resulting from soil/dust ingestion. As the concentration of As in soil exceeds 65 μg/g, the ILCR estimates resulting from incidental soil/dust ingestion become greater than HG consumption related risks. One explanation for this observation is that while the relationship between risk estimates resulting from incidental soil/dust ingestion and As soil concentration follow a linear function, the relationship between risk estimates (as a result of HG consumption) and the As soil concentration (Fig. 4) are best described by a power function ($R^2 = 0.99$).

Unlike Pb, upper percentiles of As ILCR estimates from each exposure pathway were similar in magnitude to one another over the concentration regime tested. The primary reason for this observation is the general risk assessment practice used to evaluate hazards and risks to compounds assumed to have a toxicological threshold (Pb, for the purposes of this study) versus those assumed to have no toxicological threshold (inorganic As), respectively. When predicting an ILCR estimate, an individual's average daily exposure over a lifetime, inclusive of all life-stages (infant, toddler, child, adolescent, and adult) should be developed. In the current study, the soil/dust ingestion rate of a lifetime receptor was 30 mg/d (at the 95th percentile) compared to that of the toddler (aged 6 months to 4 yr), used to develop Pb HQ values, of 202 mg/d (at the 95th percentile).

Incidental soil ingestion rates, particularly among young children, have the potential to greatly influence chemical exposure estimates and hence, the overall health-based soil standards or criteria. One of the original questions in which this work attempted to address was: does the consumption of home grown vegetables represent a significant exposure pathway for As and Pb relative to direct incidental ingestion of soil? Hence the original intent of this work was to, in part, compare and contrast exposures resulting from soil ingestion and the HG pathway over a specified concentration regime. However, future screening tools may consider combining these two pathways together.

4.3. Uncertainties and limitations

Although this screening tool has employed a 2-D Monte Carlo approach to examine (quantitatively) the inter-variability and parameter uncertainty of human health risks, there are a number of other uncertainties, limitations and assumptions that should be identified:

4.3.1. Exposure characterization assumptions

- When predicting ILCR estimates, an individual's average daily lifetime exposure must be approximated. The current study reflected a residential lifetime receptor (an individual who spends 33 yr at one residence from birth through to adulthood). It is therefore important to recognize that the residential lifetime receptor's exposure was amortized over an expected lifespan of 75 yr (Health Canada, 2004a). A more recent Draft Guidance on Human Health Preliminary Quantitative Risk Assessment (Health Canada, 2009a) recommends an expected lifespan of 80 yr.
- The PDF describing chronic soil intake rates was based on a short-term soil intake study involving 64 children. The empirical distribution provided by Stanek et al. (2001), while an improvement over point estimate values, has drawbacks in that reported intake rates at lower percentiles (1st through 7th) were associated with negative values. In general, negative soil ingestion estimates are not reset to zero since a bias may be introduced in the final soil ingestion estimate due to the inclusion of only the positive intake estimates (Stanek and Calabrese, 1995).
- Although characterized as variability within a given population, the lack of knowledge associated with the long-term consumption rates of HG vegetables among individuals was considered significant. Canadian guidance on conducting risk assessment typically employs point-estimate percentages of the total fruit and vegetable intake that might be home-grown. Much of the intake data collected by Health Canada (2004a) were based on a 24-hour recall study during the 1972–1973 National Food Consumption Survey (NFCS) and, therefore do not reflect long-term food consumption patterns of an individual, but rather the variability of reported consumption rates of many individuals over a 24-hour recall period. The U.S. EPA data used in the current assessment attempted to account for the variability in home-garden vegetable intake rates across seasons; however, the data are highly skewed.
- Default relative absorption factors for Pb and inorganic As in soil were used to adjust for the bioaccessibility in soil relative to that in the medium used to deliver the trace element in the toxicological study used to quantify hazard and/or risk. It should be recognized that the bioaccessibility of Pb and inorganic As in soils can vary significantly among sites and is dependent on a number of site-specific factors (such as source of contamination, soil characteristics, etc.), potentially impacting the relative difference in exposure estimates between the two exposure pathways.

4.3.2. Toxicological characterization

For the purposes of the current study, point-estimate toxicity data were selected that were consistent with Health Canada's Toxicological Reference Values (Health Canada, 2004b). The current study used a provisional tolerable intake for Pb of 3.57 μg/kg/d based on the provisional tolerable weekly intake (PTWI) recommended by the Joint FAO/WHO Committee on Food Additives (WHO, 1993). This paper does not critically evaluate the studies used to support the recent conclusions drawn by Health Canada (2011a,b) and others but rather attempts to answer the question: "does the consumption of home grown vegetables represent a significant pathway for As and Pb relative to direct incidental ingestion of soil?". In this context (*i.e.*, a relative comparison between predicted health risks from two exposure pathways), the Pb TRV applied will not change the result, so long as the TRV applied to each exposure pathway in question is the same.

- There is considerable uncertainty associated with the development of any toxicological reference dose (RfD), whether it is developed from laboratory animal or human epidemiological studies. These uncertainties have not been quantified in the current study. Similarly, the literature surrounding the human health effects related to inorganic As exposure is vast; however, the uncertainties inherent in the oral cancer slope factor for inorganic As may span up to several orders of magnitude.

4.3.3. The 2 dimensional MCA

- As suggested by the Volume III of the U.S. EPA Risk Assessment Guidance for Superfund (U.S. EPA, 2001), probability distribution functions (PDFs) assigned to characterize variability (*i.e.*, the inner loop of the 2-D MC simulation in this case) were sampled 10,000 times. This, according the U.S. EPA (2001), is typically considered sufficient to resolve any numerical stability issues at the tails of each distribution. No specific guidance or recommendations with respect to the minimum number of iterations that should be used for the outer loop of a 2-D simulation were identified. It is noted, however, that the U.S. EPA (2001) indicates that computational power can be a limiting step for some 2-D simulations. Due to the number of simulations evaluated, the computational power, and memory available, the current study employed 25 iterations in the outer loop of each 2-D simulation. The test case scenario employed 200 iterations and produced slightly different confidence limits for the same scenario (using 25 iterations), indicating that some numerical instability may be present.

5. Conclusions

This study provides an answer to the question: "does the consumption of home grown vegetables represent a significant pathway for As and Pb relative to direct incidental ingestion of soil?". The current study concluded that, for high-quantity consumers of HG vegetables (the upper 95th percentile of consumers in the general population), the HG vegetable consumption pathway can be as significant as incidental soil and dust ingestion for inorganic As and, therefore, should be considered when developing generic health-based soil criteria in residential settings. Predicted Pb HQ estimates among young children resulting from HG vegetable consumption were 4- to 10-fold lower than exposures resulting from direct incidental soil and dust ingestion. It is recognized, however, that the difference in relative risks resulting from HG vegetable consumption versus soil/dust ingestion between Pb and inorganic As is greatly influenced by the general risk assessment practice used to evaluate compounds assumed to have a toxicological threshold (for the purposes of this study Pb) versus compounds assumed to have no toxicological threshold (inorganic As). In other words, the difference in soil/dust ingestion rates used to characterize young children (the 95th percentile of 202 mg/d) versus a lifetime residential receptor (the 95th percentile of 30 mg/d) was a primary factor contributing to the relative differences observed between HQ and ILCR resulting from these two exposure pathways for Pb and inorganic As, respectively.

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